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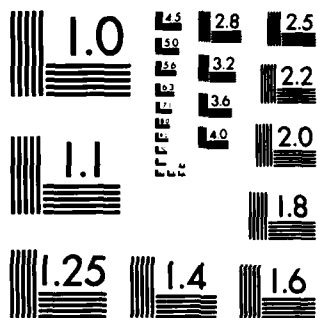
NITROGEN REMOVAL IN WASTEWATER PONDS(U) COLD REGIONS
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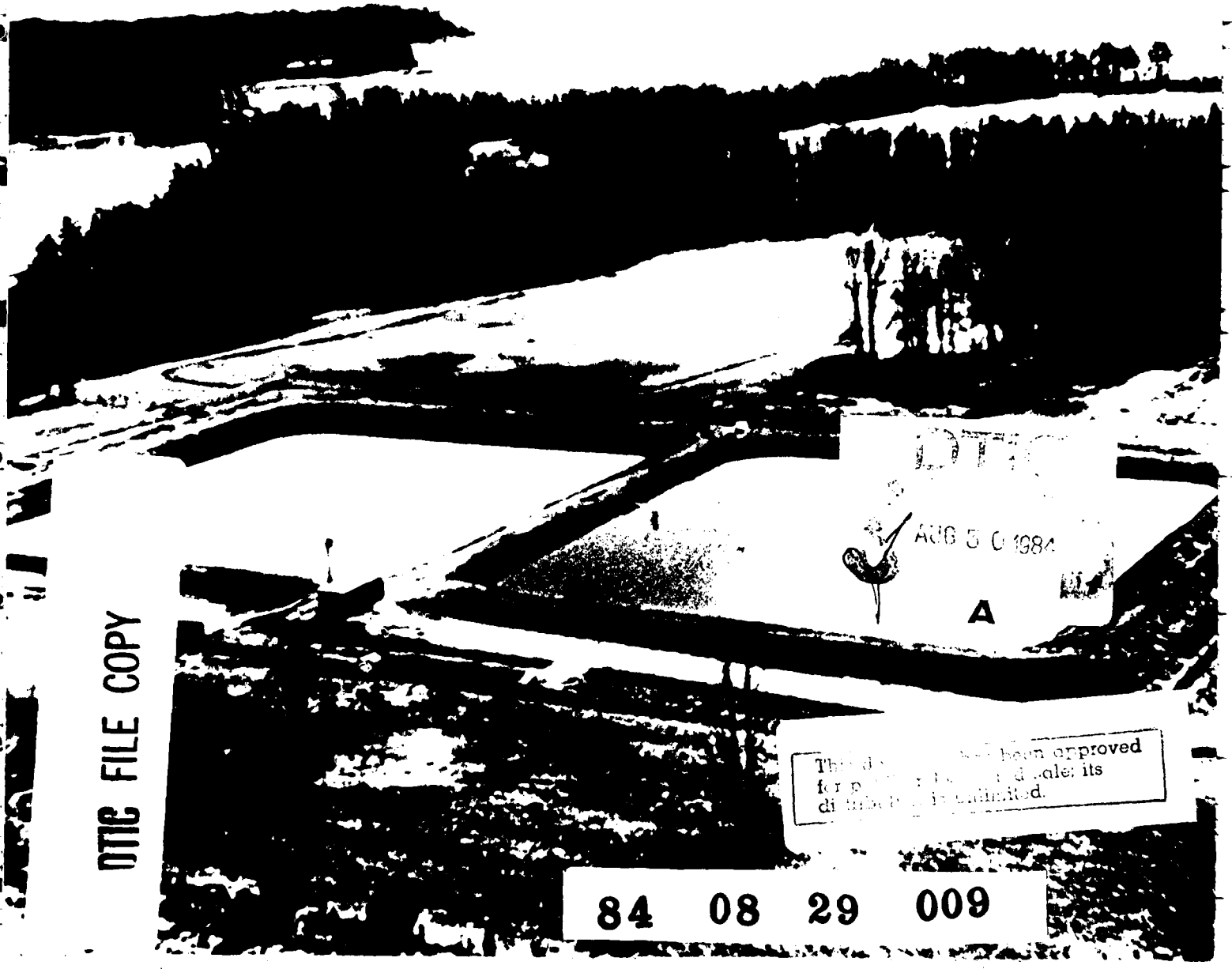


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Nitrogen removal in wastewater ponds



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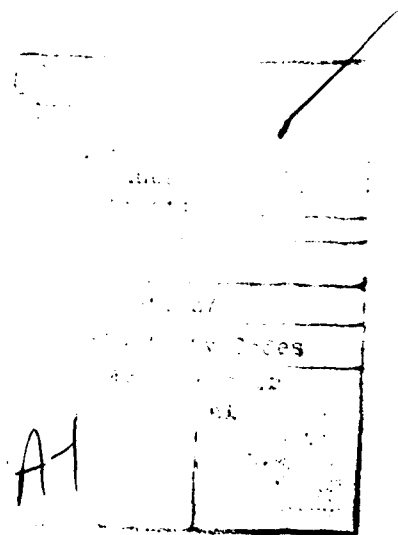
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Nitrogen removal in wastewater ponds

Sherwood C. Reed



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REPORT DOCUMENTATION PAGE		READ INSTRUCTIONS BEFORE COMPLETING FORM
1. REPORT NUMBER CRREL Report 84-13	2. GOVT ACCESSION NO. AD-A144971	3. RECIPIENT'S CATALOG NUMBER
4. TITLE (and Subtitle) NITROGEN REMOVAL IN WASTEWATER PONDS		5. TYPE OF REPORT & PERIOD COVERED
		6. PERFORMING ORG. REPORT NUMBER
7. AUTHOR(s) Sherwood C. Reed		8. CONTRACT OR GRANT NUMBER(s)
9. PERFORMING ORGANIZATION NAME AND ADDRESS U.S. Army Cold Regions Research and Engineering Laboratory Hanover, New Hampshire 03755		10. PROGRAM ELEMENT, PROJECT, TASK AREA & WORK UNIT NUMBERS DA Project 4A762730AT42 Task C, Work Unit 009
11. CONTROLLING OFFICE NAME AND ADDRESS Office of the Chief of Engineers Washington, D.C. 20314		12. REPORT DATE June 1984
		13. NUMBER OF PAGES 33
14. MONITORING AGENCY NAME & ADDRESS (if different from Controlling Office)		15. SECURITY CLASS. (of this report) Unclassified
		15a. DECLASSIFICATION/DOWNGRADING SCHEDULE
16. DISTRIBUTION STATEMENT (of this Report) Approved for public release; distribution unlimited.		
17. DISTRIBUTION STATEMENT (of the abstract entered in Block 20, if different from Report)		
18. SUPPLEMENTARY NOTES		
19. KEY WORDS (Continue on reverse side if necessary and identify by block number) Nitrogen Waste management Wastes (sanitary engineering) Waste treatment Waste water		
20. ABSTRACT (Continue on reverse side if necessary and identify by block number) Nitrogen removal from wastewater can be required in a number of situations, and many military facilities have been or will be retrofitted for this purpose. Treatment lagoons and holding or storage ponds are a common treatment method or a common component in many systems. Qualitative observations over several decades document nitrogen losses from these systems due to a variety of possible biochemical interactions. This analysis is based on an extensive body of quantitative data recently published by the U.S. EPA. A mathematical model was developed and validated that indicated that nitrogen removal from pond systems is dependent on pH, temperature, and detention time. The specific biochemical factors could not be isolated, but the analysis suggests that volatilization of ammonia is the major pathway for nitrogen loss. The model can be used as a design equation for new facilities, for retrofits, and for land treatment systems with storage ponds, since nitrogen is a critical design parameter in these cases.		

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PREFACE

This report was prepared by Sherwood C. Reed, Environmental Engineer, Civil Engineering Research Branch, Experimental Engineering Division, U.S. Army Cold Regions Research and Engineering Laboratory. Funding for this effort was provided by DA Project 4A762730AT42, *Design, Construction and Operations Technology in Cold Regions*; Task C, *Cold Regions Base Support; Maintenance and Operations*; Work Unit 009, *Water Supply and Wastewater Treatment at Cold Regions Facilities*.

The report was technically reviewed by Dr. E.J. Middlebrooks, Clemson University, Clemson, South Carolina, and R.W. Crites, G.S. Nolte & Associates, Sacramento, California.

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SUMMARY AND CONCLUSIONS

This study has produced a design equation that can be used to estimate nitrogen removal in wastewater pond systems. The equation was developed and validated with data from full-scale systems in various parts of the United States and Canada. The ranges of the parameters included in the development and validation were:

Wastewater nitrogen	8 to 61 mg/L
Detention time	5 to 231 days
Pond temperature	1° to 28°C
Pond pH	6.4 to 9.5

As a result the design equation should be valid for wastewater ponds located anywhere in the temperate, subarctic or arctic zones.

The equation can be used in those situations where nitrogen removal and/or conversion is required prior to discharge. It will produce an estimate of nitrogen removal that will occur in a pond system for cost effectiveness comparison with alternative removal methods.

The equation should also be used in the design of all land treatment systems that incorporate a pond for treatment or storage. Nitrogen is typically the controlling parameter in system size and therefore cost, and a reliable estimate of nitrogen removal in the pond will allow more cost-effective design of the land treatment component.

As a result of this study it is concluded that:

1. The nitrate concentration in the effluent from non-aerated and partial mix aerated ponds will be low. Essentially all of the nitrogen in the effluent will be in either the ammonia or organic form.

2. The nitrogen removal in these pond systems is dependent on pH, temperature and detention time. Under warm, sunny conditions with adequate detention and alkalinity, nitrogen removal

can exceed 80%. Although the remaining nitrogen will not be in the nitrate form, it may still be possible to meet discharge standards because of the high overall removal.

3. The dependence on pH, temperature and detention time suggests that volatilization (loss of gaseous ammonia to the atmosphere) is the major factor responsible for long-term permanent nitrogen removal from pond systems. However, there is no direct, absolute proof to support this hypothesis.

4. The loss of gaseous ammonia to the atmosphere from beneath a continuous ice cover seems unlikely. However, nitrogen removal was observed under such conditions in the systems in the northern United States and Canada that were included in this study. This suggests that some other mechanism may prevail in the winter, with volatilization again occurring when conditions are favorable.

5. The presence of an ice cover on a wastewater pond restricts oxygen transfer and results in an increase of dissolved constituents in the remaining liquid due to rejection during the freezing process. An equation to estimate the effect was developed in this study. All of these factors combined mean that ice-covered, non-aerated ponds will produce very poor effluent quality during the period of ice cover.

6. Methods for estimating the pH and the temperature in pond systems are presented in this report. Such data are essential to determining nitrogen removal and are useful for other purposes as well.

7. Use of the design equation should ensure maximum cost effectiveness for design of discharging systems that require nitrogen removal or conversion, and for all land treatment designs where nitrogen is the controlling parameter.

NITROGEN REMOVAL IN WASTEWATER PONDS

Sherwood C. Reed

INTRODUCTION

Wastewater ponds are a relatively common and cost-effective treatment method. Labor and energy requirements are low compared to other mechanical treatment systems, and the concept is particularly attractive for military facilities, where sufficient land is usually available.

Ponds can be used as the sole treatment method with discharge to a nearby watercourse, as a final polishing step after other forms of treatment, or as preliminary treatment/storage prior to land application of wastewater. Ponds are also classified with respect to the degree of mechanical mixing or aeration provided. Table 1 summarizes the characteristics of the pond systems in common use.

The forms and transformations of wastewater nitrogen are critical to engineering design and water quality decisions. Nitrogen in any form may eventually promote eutrophication in receiving waters, and in some cases its significant reduction or total removal may be required prior to discharge. In many situations, the presence of ammonia or ammonium ions is critical, since their

oxidation will reduce the oxygen content of the receiving waters. In addition, ammonia is toxic to a wide variety of fish in relatively low concentration,²² and the U.S. Environmental Protection Agency recommends a limit of 0.02 mg/L based on its impact on salmonid fish fry. As a result, many states limit the discharge of ammonia nitrogen at least during the spring and summer months, when removal is required prior to discharge. Nitrogen is also the limiting design parameter for many land treatment systems; therefore, the concentration of nitrogen leaving the preliminary treatment/storage pond is a critical factor in determining the size and therefore the cost of the overall system.

It is still a common engineering practice to ignore any nitrogen transformations or removal that might occur in a pond system. It has been observed on numerous occasions that nitrogen losses do occur in pond systems, but there were insufficient quantitative data to permit the development of a rational design approach. The recent publication^{1 10 16 21} of very detailed pond studies documents nitrogen removal of over 80% under favor-

Table 1. Design factors for treatment ponds

Type	Detention time (days)	Depth (m)	BOD loading (kg/ha day)
Oxidation	10-40	0.45-0.91	67-134
Facultative			
Winter average air temperature			
above 60 °F	25-40	0.91-1.5	45-90
32 °F-60 °F	40-60	1.2-1.8	22-45
below 32 °F	80-180	1.5-2.1	11-22
Partial mix aerated	7-20	2.4-3.0	34-112

able conditions, and there is now a sufficient body of data to develop and verify a mathematical description. This report presents the development of a rational design approach. Use of this approach will permit the estimation during design of the nitrogen concentration to be expected in a pond effluent, and thereby ensure cost effectiveness in design, construction and operation of any subsequent treatments that may be included.

BACKGROUND

The loss of ammonia nitrogen from streams and from alkaline water impoundments was described by Stratton in 1968-69.^{23,24} He suggested a first-order reaction to describe the loss and indicated that the rate constant was proportional to pH, temperature, detention time and the degree of air movement and water turbulence at the water surface. He estimated that the ammonia losses from a treated wastewater impoundment in Santee, California, approached 6% per day.²⁴

In a 1968 summary of nitrogen removal methods, Johnson¹² suggested that the removal of nitrogen in wastewater ponds is dependent on the growth and harvest of algae and on soil adsorption of ammonium ions if seepage is allowed. Algal harvesting was considered essential by the investigators he cited. Otherwise the algae would die and decompose, and the nitrogen would be recycled to the pond or to the receiving stream. Pilot studies by DiGiano indicated that algal uptake, not nitrification, was the major factor for the nitrogen transformations he observed.³ Porcella et al.,¹⁹ in a study of a tertiary effluent impoundment in Alpine County, California, noted that 48% of the influent nitrogen over a 2-year period could not be accounted for, and they suggested nitrification-denitrification as the most plausible reason. They also noted algae growth and suggested that trout mortality was due to ammonia toxicity.

Ammonia removals of about 70% were reported by Mara et al.¹⁴ for a five-cell pond in Brazil (average temperature 25°C, median pH 8, detention time 29 days), but removal mechanisms were not discussed. Very significant nitrogen removals were measured in the four EPA studies previously mentioned^{1, 10, 16, 21} but no discussion of removal mechanisms was presented. Ferrara and Harleman⁷ developed a mathematical model (complete mix—dynamic) that indicated that the major pathway for nitrogen removal in facultative ponds was biological and algal uptake followed by loss to

bottom sediments. King¹¹ reported on nitrogen removal in a pond system in Michigan, and indicated that the harvest of aquatic plants (algae, etc.) could only account for about 10% of the nitrogen removed. He achieved up to 95% nitrogen removal in the 120-day detention time, and suggested that loss of ammonia, as a gas, to the atmosphere was the major factor. He presented the following first-order equation to describe the results achieved:

$$N_t = N_0 e^{-0.03t}$$

where N_t = nitrogen (total, as N) at time t
(mg/L)

N_0 = nitrogen (total, as N) entering pond
(mg/L)

t = detention time (days).

Idelovitch¹¹ described very high ammonia removals from ponds in Israel during the period 1975-79. The pH was first raised, by lime treatment, to about 11, and ammonia removals of 95% were observed after 14 days detention, with air stripping identified as the responsible mechanism. In addition to the alkalinity/pH requirements, Idelovitch indicated that removal efficiency is also dependent on water temperature and detention time.

It is clear from the review presented above that nitrogen losses from pond systems have been recognized for a number of years, but that there has been no consensus on identification of the responsible mechanisms. The publication of extensive new data^{1, 10, 16, 21} from actual pond systems has not yet resolved the issue. Pano and Middlebrooks¹⁸ developed a complete-mix mathematical model using data from the first cell in three of the EPA pond studies^{1, 16, 21} and then applied the model to the results from cells 2-4 in these same systems for verification. Independent data from other full-scale operating systems were also found to fit the model. Their relationship is dependent on pH, temperature and hydraulic loading rate, and air stripping of ammonia is suggested as the major removal pathway. Ferrara and Avci⁶ applied a model previously developed at MIT⁷ to some of the same data²¹ used by Middlebrooks and Pano, but concluded that ammonia volatilization (air stripping) accounted for a very small fraction of the total nitrogen removed. They suggested biological and algal uptake followed by loss as sediment as the major pathways.

This continuing controversy may not be capable of resolution with the present data base. The tem-

perature, pH and detention time conditions that might result in high rates of ammonia volatilization would also support high rates of biological activity and algal uptake of ammonia. It may not be possible to determine whether the Pano and Middlebrooks model or the Ferrara and Avci model is the more valid description of nitrogen removal. Additional concerns arise since both are based on a complete-mix assumption, which is not entirely valid for the long detention time, multiple-cell facultative ponds considered. In addition, the Pano and Middlebrooks equation deals only with the ammonia fraction entering a pond and ignores the other nitrogen forms that may be present. These other forms may then cycle through several stages and appear as ammonia while the wastewater is in the pond. The Ferrara and Avci equations assume a very significant nitrogen loss to bottom sediments. Assumptions regarding the permanence of this disposition seem optimistic and tend to minimize the solubility of nitrogen and to ignore the resuspension of particulate matter caused by density currents and/or seasonal "overturns" due to thermal effects in a pond.

THEORY

1. Nitrogen forms

Nitrogen can exist in wastewaters in a number of different forms because of the various oxidation states represented, and it can readily change from one state to another, depending on the oxidation or reduction reactions induced by the physical or biochemical conditions present at the time. The principal forms of concern in wastewater are:

Organic nitrogen: Organic nitrogen is present in the protein molecules in animal tissue and in fecal matter (typical assumption is about 8–10% nitrogen content). Microorganisms decompose the organic nitrogen and produce ammonia (NH_3 , NH_4^+).

Ammonia: Ammonia can be present as molecular ammonia (NH_3) or as ammonium ions (NH_4^+). The nitrogen in urine is initially urea, which is hydrolyzed by an enzyme to ammonium carbonate ($(\text{NH}_4)_2\text{CO}_3$). Microorganisms (Nitrosomonas and Nitrobacter) oxidize these ammonia compounds to nitrite (an unstable intermediate) and then nitrate. Ammonia can also be taken up by either bacteria or plants and converted to cell matter, which returns the nitrogen to the organic form.

Nitrate: Nitrate (NO_3^-) is the stable end point of the oxidation reactions:

Organic N—Ammonia—Nitrite—Nitrate

Nitrate can be taken up by plants and algae and converted to organic nitrogen in cell tissue. Or, in an oxygen-deficient or anaerobic environment (with sufficient carbon present), it can be reduced to elemental nitrogen (N_2) and lost as gas. The nitrifying organisms do not compete and perform effectively until the simple carbonaceous organic materials (BOD) have been substantially removed from the wastewater. It is also believed that the nitrifying organisms function more effectively if they are attached to surfaces rather than suspended (e.g. free-floating) in the liquid column. As a result, a relatively low degree of nitrate conversion seems to occur in typical wastewater treatment systems. To achieve nitrification the system design must provide for either relatively low initial organic loadings or long detention times, sufficient oxygen and suitable temperatures, and, for maximum conversion efficiency, a surface for attached growth. As discussed in more detail later, these conditions are not simultaneously present in the typical facultative pond and the apparent rates of nitrate conversion are low.

The nitrogen concentration in typical municipal wastewaters ranges from about 15 to over 50 mg/L. About 60% of this is in the ammonia form, about 40% in the organic form.²³ To avoid confusion it is conventional to express these concentrations as equivalent elemental nitrogen (as N) regardless of the particular ionic form. For example, 10 mg/L of nitrate nitrogen (as N) is equal to about 45 mg/L of the specific nitrate compound (as NO_3^-).

The presence of ammonia gas (NH_3) or ammonium ions (NH_4^+) in the wastewater is strongly dependent on the pH of the water and on temperature. Figure 1 illustrates this relationship. At pH 7 essentially only ammonium ions are present, at pH 12 only dissolved ammonia gas. This dissolved gas can be released from the liquid under proper conditions, and this is the purpose of air-stripping operations in mechanical treatment plants. Typically, the pH is elevated with lime, the sludge is allowed to settle, and then the clarified effluent is aerated or allowed to cascade over a packed bed in a stripping tower. At the high hydraulic loading rates commonly employed the concept is only effective in warm weather because of the temperature constraints illustrated in Figure 1.

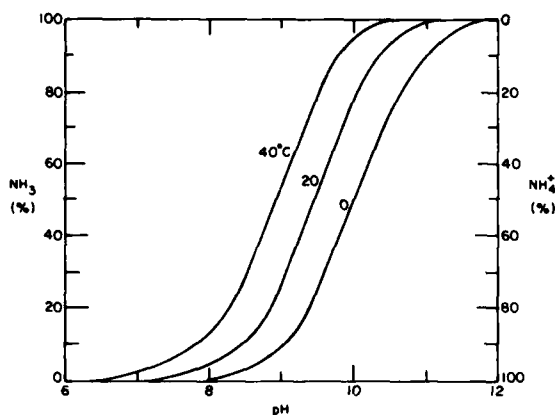


Figure 1. Ammonia-ammonium ion distribution in water—influence of pH and temperature¹⁷.

2. Pond system hydraulics

It is common in the design and analysis of suspended growth biological wastewater treatment systems to characterize the system as either completely mixed, plug flow, or intermediate. The complete-mix approach assumes that the reactor is continuously and completely mixed, so that a sample taken from any point at a particular time would be identical to a sample from any other point in the reactor. Aerated systems with high-intensity agitation approach these conditions, and the equation describing biochemical removal of a constituent typically takes the form:

$$\frac{A_f}{A_i} = \frac{1}{1 + K_c t}$$

where A_f = final concentration of constituent
 A_i = initial concentration of constituent
 K_c = first order, temperature-dependent, complete-mix reaction rate constant
 t = hydraulic residence time.

Activated sludge systems and complete-mix pond systems are designed based on this approach. Many partial-mix (see Table 1) ponds have also been designed this way, even though actual conditions do not always fit the design assumptions.

The plug flow design assumes that a unit of wastewater retains its identity throughout its passage through the system and is not mixed with other increments. A sample taken near the end of the system would not be identical in character to a sample taken near the influent point. It is therefore not appropriate to compare input and output samples taken at the same time and conclude that

a certain removal has occurred unless the input concentration does not vary with time. Since this is not the case in typical systems, the true removal can only be determined by comparing the characteristics of the final sample at discharge with its own characteristics when it entered the system. The equation describing the biochemical removal of a constituent under plug flow conditions typically takes the form:

$$\frac{A_f}{A_i} = e^{-K_p t}$$

where K_p = first-order, temperature-dependent, plug flow reaction rate constant and other terms are as defined previously.

Pond systems that do not have mechanical mixing or aeration (i.e. oxidation ponds, facultative ponds, etc.) are generally considered to be plug flow reactors.¹⁷ Wind action and thermally induced density currents can induce both vertical and lateral mixing, but these are usually transient effects compared to the long detention time in the system. Bowles and Middlebrooks² have shown that short-circuiting of flows occurs in many pond systems due to configuration, inlet and outlet location, etc. The actual detention time in the ponds studied in their work ranged from 29% to 89% of the assumed design detention time. These reductions are due to "dead spots" in the pond, so the plug flow assumption would still apply to the bulk of the liquid moving through the system.

3. Pond biochemistry

The ecosystem in plug flow wastewater treatment ponds is very dynamic. It is similar in many respects to the ecosystem found in natural ponds, but the diversity of species tends to be less and the density of those organisms present tends to be greater than the natural case because of the higher organic and nutrient loadings entering the system. Most of the physical, chemical and biochemical reactions taking place in ponds have an impact on the nitrogen present in the system.

Most of the nitrogen entering the system will be in the ammonia form, either as a dissolved gas or as ammonium ions. The remainder will be organic nitrogen associated with the solid particulate matter. Some of the latter can settle directly to the bottom due to gravity forces. Either ammonia or organic N can be consumed by animals or bacteria and then eventually settle. The ammonia forms can be taken up by plants and algae and also eventually settle. These are the major pathways suggested by Ferrara and Avci⁴ for nitrogen removal

Table 2. Data summary from EPA lagoon studies.^{1 10 16 21}

Location	Detention time, d*	Liquid temp (°C)			Average algal conc (no./mL)			Influent total N (mg/L)			Nitrogen removal (%)		
		Ann avg	Apr-Sep	Oct-Mar	Ann avg	Apr-Sep	Oct-Mar	Ann avg	Apr-Sep	Oct-Mar	Ann avg	Apr-Sep	Oct-Mar
Peterborough	107	11	18	5	398,000	216,000	563,000	17.8	15.4	20.1	43	50	38
Kilmichael	214	18.4	24.2	13.4	429,600	196,000	580,000	35.9	41.8	30.1	80	83	76
Eudora	231	14.7	02	7.2	412,178	440,000	384,000	50.8	52.2	49.3	82	80	83
Corrine	42	10	14.8	4.3	781,000	1,223,087	423,072	14.0	11.1	16.9	46	56	36

*Note: Peterborough, Kilmichael and Eudora are three-cell systems. Corrine is a seven-cell system. Detention times, temperatures, etc. are given for the first three cells at Corrine.

from pond systems. It is believed that these can be significant pathways, but there is also a strong potential for reintroduction of nitrogen into the water column from this settled material. These biological cells might contain up to 8% nitrogen, but most of the cellular material is susceptible to decomposition. It is estimated¹⁵ that the nonaerobically-biodegradable portion might range from 20% to 50%. This means that up to 80% of the settled nitrogen is again potentially available for solution in the water column. Most of this will again be in ammonia form, and any portion of that in the gaseous form is potentially available for loss to the atmosphere. The portion that remains temporarily with the bottom sediments is not absolutely permanent due to the periodic resuspension of some bottom material during seasonal overturns, the consumption of the suspended material by protozoa and other animals, and the anaerobic decomposition of the material in place.

Some qualitative indications regarding the role of the algae in the removal of nitrogen result from an examination of the data collected at the four facultative ponds in the recent EPA studies.^{1 10 16 21} Table 2 summarizes pertinent average data from these four systems and presents annual and seasonal values. The nitrogen removals shown were calculated using a plug flow assumption (i.e. the effluent leaving the pond today, in a 30-day pond, is not compared to the influent entering today, but rather to the influent that entered 30 days ago). The pond systems studied represent a wide geographical distribution, and all show significant seasonal trends with respect to liquid temperature in the system. However, the average algal concentrations *do not* show a consistent seasonal trend, although three of the four do show a significant seasonal difference for nitrogen removal.

If algae constitute the major factor responsible for permanent nitrogen removal, it then seems

likely that maximum removals should be observed during or very soon after peak algal growth periods. It is possible to test this hypothesis by examining the average monthly data from these same four systems. Figures 2-5 present the results of this analysis. The deviation from the mean annual values is plotted for both algal concentration and nitrogen removal to more clearly illustrate the trends.

The three pond systems in the colder climates (Peterborough, New Hampshire, Eudora, Kansas, and Corinne, Utah) exhibit an algal peak in spring, which is probably due to the seasonal overturn of the pond and the resuspension of benthic nutrients. Two of these (Peterborough and Corinne) also show an algal peak in fall and early winter when another overturn would be expected. (The temperature must gradually change and then remain at about 4°C for an overturn to occur. These conditions did not exist in the fall and early winter at the Eudora site during the study period, so only the spring overturn was observed.)

Higher nitrogen removals are demonstrated at all four sites during the warmer months (Eudora is marginal in this respect), but there is no apparent correlation between the periods of higher nitrogen removals and the peak periods for algal growth. This would seem to indicate that algal uptake is not the major factor responsible for permanent nitrogen removal.

The bacterial oxidation of ammonia to nitrate is thought to be minimal in pond systems, as indicated previously. The average nitrate concentrations in the same four pond systems^{1 10 16 21} are summarized in Table 3. Nitrate concentration in the effluents is very low in all of the systems and represents less than 1% of the total nitrogen that entered. There are no consistent seasonal trends indicated by the data, and one system (Corinne) shows a net decrease in nitrate on a year-round basis. These low values do not prove that nitrification is not oc-

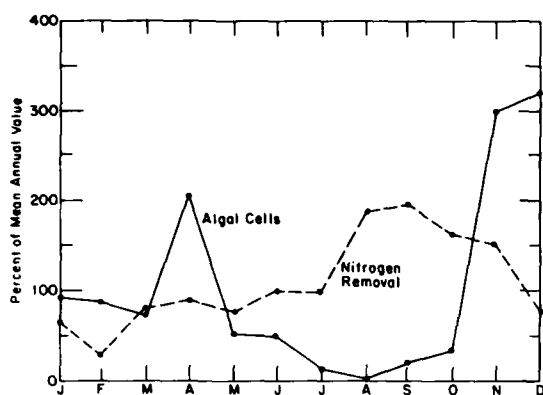


Figure 2. Algal concentration and nitrogen removal, Peterborough, New Hampshire.

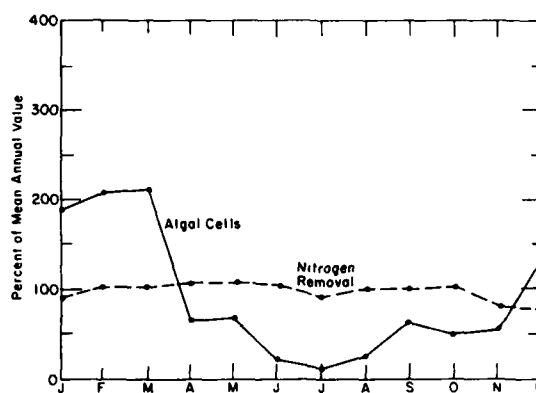


Figure 3. Algal concentration and nitrogen removal, Kilmichael, Mississippi.

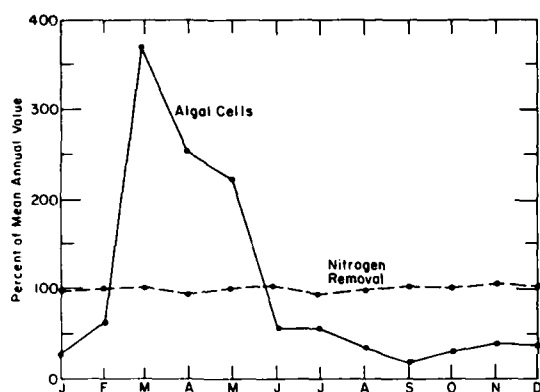


Figure 4. Algal concentration and nitrogen removal, Eudora, Kansas.

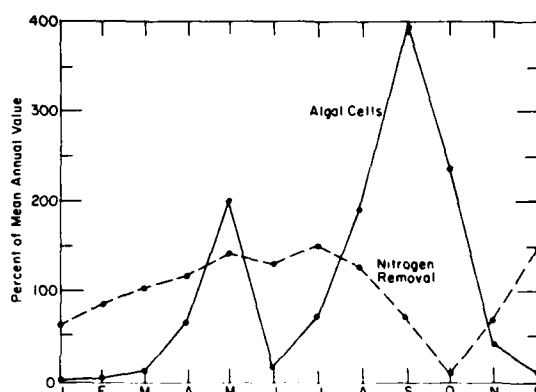


Figure 5. Algal concentration and nitrogen removal, Corinne, Utah.

curing in ponds. It may be possible that significant nitrification is taking place, but the nitrates are then taken up by the algae and/or denitrified and lost as nitrogen gas. However, it appears more likely that nitrification rates are low in pond systems since the responsible organisms require oxygen and relatively low levels of biodegradable organics, and are thought to prefer attachment to surfaces. These three factors do not simultaneously exist to a significant degree anywhere in a typical pond. (High levels of nitrification were observed in an aquaculture pond in Hercules, California, where suspended plastic strips were included as a support medium for biological growth.²⁰)

If significant nitrification is occurring in typical ponds, it can only be an intermediate stage, as evi-

denced by the low nitrate concentrations in the system effluents. The permanent removal of nitrate would then require biological uptake and settling and/or denitrification. Denitrification requires an anaerobic environment and an adequate carbon source. Both of these factors will exist at the pond bottom, but the presence of significant nitrates at the same location seems unlikely. Particulate matter and dead vegetation, etc., may settle and become part of the benthic deposit, but the nitrogen tied up in those materials would be in the organic form, and their subsequent oxidation all the way to nitrate seems unlikely in the anaerobic environment present on the bottom.

If algal uptake/settling, nitrification, and denitrification are eliminated as major pathways for permanent nitrogen removal, the only possibility

Table 3. Nitrates in facultative ponds.

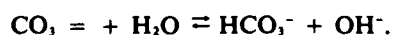
Location	Nitrite plus nitrate (mg/L)			
	Influent, annual average	Effluent*		
		Annual average	April- Sept	Oct- Nov
Peterborough	0.10	0.19	0.22	0.17
Kilmichael	0.11	0.19	0.13	0.25
Eudora	<0.1	0.67	0.55	0.78
Corinne	0.20	0.04	0.04	0.04

*This is final effluent from the three-cell systems at Peterborough, Kilmichael and Eudora, and is effluent from the third cell at Corinne.

left is air stripping or volatilization of ammonia. The equilibrium equation for ammonia in water is represented by:



Figure 1 illustrates the relationship at various temperatures. At typical ambient temperatures and low hydrogen ion concentrations (pH 10.5 +), essentially all of the ammonium is in gaseous form. A high pH is achieved in mechanical treatment plants by adding lime or similar chemicals. Elevated pH values are also achieved in ponds due to the interactions of algae and the carbonate buffering system. The latter is represented by the following equations:



During algal photosynthesis carbon dioxide (CO_2) is removed from solution. The result is to decrease both the bicarbonate alkalinity and the hydrogen ion concentration (increases pH). Extreme diurnal fluctuations in the pH are observed in many ponds because of this relationship between pH and algal photosynthesis. On warm, sunny days pH values exceeding 10-11 are common in the surface layers of many ponds. Oxygen is another product of algal photosynthesis since the carbon is extracted from the CO_2 and the elevated pH is typically accompanied by supersaturated oxygen conditions. King¹¹ has shown that the rates of solution of atmospheric carbon diox-

ide are too slow to support optimum photosynthesis, so the algae draw on the CO_2 present in the alkalinity system. This suggests that bicarbonate alkalinity has an influence on the air stripping of ammonia and that wastewaters low in alkalinity might not be as responsive in pond systems.

A pH of least 10-12 is considered essential for efficient ammonia air stripping in mechanical systems. The near-surface bulk liquid in ponds may only approach that level for a brief period, and then only on warm, sunny days. However, the detention time in the mechanical system is measured in terms of minutes or hours, while in the pond it is many days or months. Even if the pH is only 8 or slightly above, and the gaseous fraction of ammonia stays less than 10%, the long detention time can provide a sufficient period for effective removal. Even if the pH of the bulk liquid is close to neutral, it seems possible that there could be microsites at the surface with an elevated pH due to algal activity resulting in gaseous losses over the long term.

The long detention period also provides enough time for the nitrogen entering a pond system to cycle through a number of oxidation states and/or cycles of plant uptake and decomposition. Every time a cycle is completed the nitrogen goes through the ammonia form and a portion is available for loss as a gas. It seems very unlikely that the nitrogen entering a pond will remain in the same discrete form throughout its passage through the system. It is far more likely that one or more transformations will occur so that the nitrogen forms leaving a pond are a result of the activity in the pond and not a residue of the incoming material. Therefore, it seems inappropriate to compare a particular form in the effluent to the same material in the influent and conclude that some absolute level of removal has occurred.

For engineering purposes it seems both prudent and conservative to consider all of the total nitrogen entering and leaving the system, and then to assume that essentially all of the nitrogen leaving the system is still in the ammonia form. There is a consensus among recent investigators that nitrogen removal in ponds is dependent on at least pH, temperature and detention time, even though there is dispute over the specific mechanism involved. In the opinion of this author, the volatilization pathway seems more likely, but there is no absolute proof available. Fortunately, it is not necessary to resolve the issue in order to develop a rational design procedure.

DEVELOPMENT OF MATHEMATICAL MODEL

1. General consideration and data base

Based on the factors discussed above, a first order plug flow equation was adopted as the appropriate model to describe nitrogen removal in pond systems. The equation takes the general form:

$$N_e/N_0 = e^{-Kf(T, t, \text{pH})} \quad (1)$$

where N_e = total nitrogen in pond effluent at the end of the detention period (mg/L)

N_0 = total nitrogen in pond influent at the start of the detention period (mg/L)

K = first-order reaction rate constant, a function of temperature (T), time (t), and pH.

The data used to develop this model were obtained from the four EPA pond studies previously mentioned:^{1, 10, 16, 21}

Peterborough, New Hampshire. The system is a three-cell facultative pond, operated in series, with a continuous discharge. During the study period the mean flow was 1011 cubic meters per day (0.27 million gallons per day). Based on this mean flow the hydraulic detention period was 107 days. The design depth in all cells was 1.2 m (4 ft), and the total surface area for the three cells was 8.5 hectares (21 acres).²¹

Kilmichael, Mississippi: A three-cell facultative pond, operated in series, with a continuous discharge. During the study period the mean flow was 280 m³/d (0.07 mgd). Based on this mean flow the hydraulic detention period was 214 days. The design depth was about 2 m (6.6 ft) in all cells, and the total surface area for the three cells was 3.3 ha (8.1 acres).¹⁶

Eudora, Kansas: A three-cell facultative pond, operated in series, with a continuous discharge. During the study period the mean flow was 500 m³/d (0.13 mgd). Based on this mean flow the hydraulic detention period was 231 days. The design depth was 1.5 m (5 ft) and the total surface area for the three cells was 7.8 ha (19.3 acres).¹⁰

Corinne, Utah: A seven-cell facultative pond, operated in series, with a continuous discharge. Only the first three cells were utilized in this analysis. Based on the mean flow of 690 m³/d (0.18 mgd) during the field study the detention period in the first three cells was 42 days. The design depth was 1.2 m (4 ft) and the surface area was 2.3 ha (5.75 acres) in these first three cells.¹

All four systems receive raw municipal (essentially domestic) wastewater from the adjacent community. Data collection continued for a full year at each site to include all seasonal influences. An ice cover was noted at both Peterborough and Corinne during some winter months. The averaged data values used for this analysis are tabulated in Appendix A. The nitrogen concentrations given are total nitrogen values, and are obtained by summing the TKN (total Kjeldahl nitrogen) and the NO₂-NO₃ (nitrite-nitrate) data. Since the TKN includes both the organic N and the ammonia N fractions, it is a close approximation of the total nitrogen present in untreated wastewater.

The effluent nitrogen values listed in the tables in Appendix A are the actual average monthly values measured for the month indicated. All of the other data listed (pH, temperature, influent nitrogen, etc.) were obtained by applying the plug flow assumption to the raw data. For example, in the 42-day detention time pond at Corinne, the effluent nitrogen listed for January is the average value for the period 1-31 January. However, the influent values listed represent the characteristics of the January effluent when it entered the system, so they are an average of the influent measurements for the period 20 Nov-20 Dec. Similarly, the temperature and pH values listed are the average of measurements made in the pond during that particular 42-day detention period. For this iteration no adjustments were made in the detention times used, to compensate for variations in incoming flow and/or water losses due to seepage and evaporation. These factors could be very significant where sustained seasonal peak flows occur or in arid climates with a high evaporative loss.

Each monthly value tabulated in Appendix A represents the average of at least five to eight measurements made during the period of concern. Since the ponds all had multiple cells and the inputs and outputs were measured for each, it is possible to arrange the data in a number of configurations for analysis. Table 4 lists the cases selected for development of the model. The total data base for each parameter is at least 1000 points, since each case has 12 monthly values and each value is an average of five to eight measurements. The detention times represented range from 27 to 231 days, the temperature from 1.2° to 26.3°C, and pH from 6.4 to 9.5. It was possible to identify specific periods when the other variables were relatively constant for analysis of the effect of detention time, temperature and pH. With one exception (Case 9), all of the data sets represent treatment performance, starting with raw untreated

Table 4. Data configurations used for model development.

Peterborough

Case 1 (Table A1) All three cells, $t = 107$ days, input to cell 1, output from cell 3, pH and T are average of all three cells.
Case 2 (Table A2) First cell, $t = 44$ days, input to cell 1, output from cell 1, pH and T are average of cell 1.
Case 3 (Table A3) First two cells, $t = 74$ days, input to cell 1, output from cell 2, pH and T are average of cells 1 and 2.

Kilmichael

Case 4 (Table A4) All three cells, $t = 214$ days, input to cell 1, output from cell 3, pH and T are average of all three cells.
Case 5 (Table A5) First cell, $t = 136$ days, input to cell 1, output from cell 1, pH and T are average of cell 1.
Case 6 (Table A6) First two cells, $T = 191$ days, input to cell 1, output from cell 2, pH and T are average of cells 1 and 2.

Eudora

Case 7 (Table A7) All three cells, $t = 231$ days, input to cell 1, output from cell 3, pH and T are average of all three cells.
Case 8 (Table A8) First cell, $t = 93$ days, input to cell 1, output from cell 1, pH and T are average of cell 1.
Case 9 (Table A9) Second cell, $t = 44$ days, input to cell 2, output from cell 2, pH and T are average of cell 2.
Case 10 (Table A10) First two cells, $t = 137$ days, input to cell 1, output from cell 2, pH and T are average of cells 1 and 2.

Corinne

Case 11 (Table A11) All three cells, $t = 42$ days, input to cell 1, output from cell 3, pH and T are average of all three cells.
Case 12 (Table A12) First cell, $t = 27$ days, input to cell 1, output from cell 1, pH and T are average of cell 1.
Case 13 (Table A13) First two cells, $t = 34.5$ days, input to cell 1, output from cell 2, pH and T are average of cells 1 and 2.

sewage since that is the purpose of the model. Case 9 was developed for comparison with Cases 2 and 11 so there would be additional data at essentially the same detention time for analysis of pH influences.

2. Detention time and temperature

Selected data were organized into groups that reflected similar pH and temperature conditions. Four such groups could be developed at 4°C, 8°C, 18°C and 24°C respectively. Figure 6 illustrates the 18°C group, which had a pH range from 8.3 to 8.4. The equation of best fit takes the form:

$$N_e/N_0 = e^{-K_T(t+A)} \quad (1)$$

where K_T = temperature-dependent rate constant
 t = detention time (days)
 A = constant, related to pH.

If nitrogen removal were only dependent on detention time then A would be equal to zero and N_e/N_0 would equal 1 when t equals zero. The slope of the line in Figure 6 accurately describes the time-dependent rate (correlation coefficient 0.98, significant at 0.01 level). The time dependence is described by:

$$N_e/N_0 = e^{-K_T(t)} \quad (2)$$

The rate constants at 4°C, 8°C and 24°C were also determined as described above, and are listed in Table 5 and plotted in Figure 7.

The general expression used to describe the temperature dependence of biological wastewater treatment processes is:

$$K_T = K_{20}(\theta)^{(T-20)} \quad (3)$$

where K_T = rate constant at temperature T
 θ = temperature coefficient
 T = temperature (°C).

A regression analysis of the data in Figure 7 produces:

$$K_T = 0.0064 (1.039)^{(T-20)} \quad (4)$$

where $K_{20} = 0.0064$
 $\theta = 1.039$
(correlation coefficient 0.94, at 0.06 level).

The general equation for time dependence becomes:

$$N_e/N_0 = e^{-0.0064t} \text{ (at } 20^\circ\text{C)}. \quad (5)$$

Equation 4 can then be used to determine the appropriate value of K_T at temperatures other than 20°C.

Table 5. $K_{T(T)}$ constant vs temperature.

Temperature (°C)	K
4	0.0031
8	0.0050
18	0.0065
24	0.0068

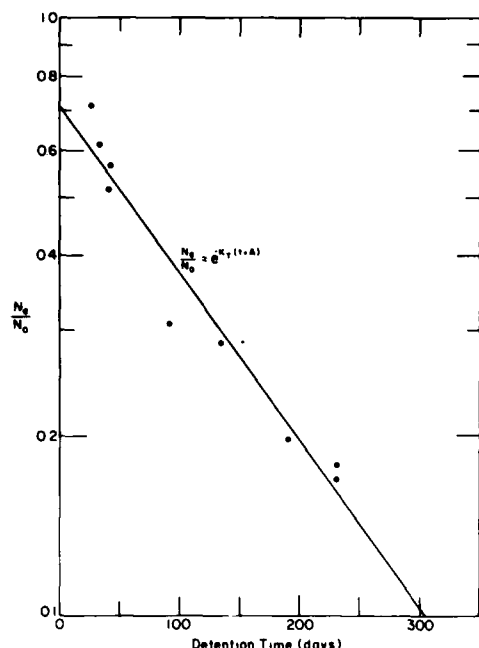


Figure 6. N_e/N_0 vs detention time. Data from Kilmichael, Eudora and Corinne, for similar temperature (18°C) and pH conditions (8.3–8.4).

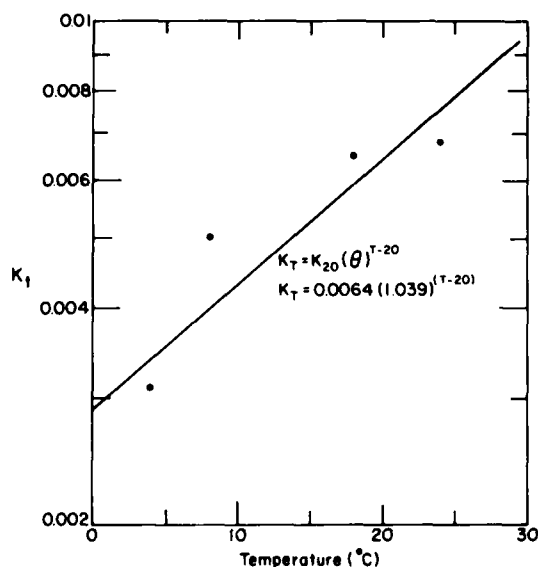


Figure 7. Time-dependent rate constant vs temperature.

3. pH dependence

It is possible to determine the pH influence independently by comparing data groups at essentially the same detention time and temperature (Cases 2, 9 and 11). Figure 8 illustrates a typical set at an average temperature of 16.5°C . These data can be described by:

$$N_e/N_0 = e^{-0.313(\text{pH}-6.6)} \quad (6)$$

The correlation coefficient was 0.86 and is significant at the 0.05 level, indicating that a good approximation is provided.

A temperature dependence was indicated by the data, so rate constants were determined for other temperature conditions within this 42–44 day detention time group. These results are plotted in Figure 9, and the data can be described reasonably well with the same theta value used in equation 4:

$$K_{\text{pH}(T)} = K_{\text{pH}(20)}(\theta)^{(T-20)} \quad (7)$$

where $K_{\text{pH}(T)}$ = pH rate constant at temperature T ($^\circ\text{C}$)

$$\begin{aligned} K_{\text{pH}(20)} &= \text{pH rate constant at } 20^\circ\text{C} \\ &= 0.388 \\ \theta &= 1.039. \end{aligned}$$

Therefore, at 20°C eq 6 becomes:

$$N_e/N_0 = e^{-0.388(\text{pH}-6.6)} \quad (8)$$

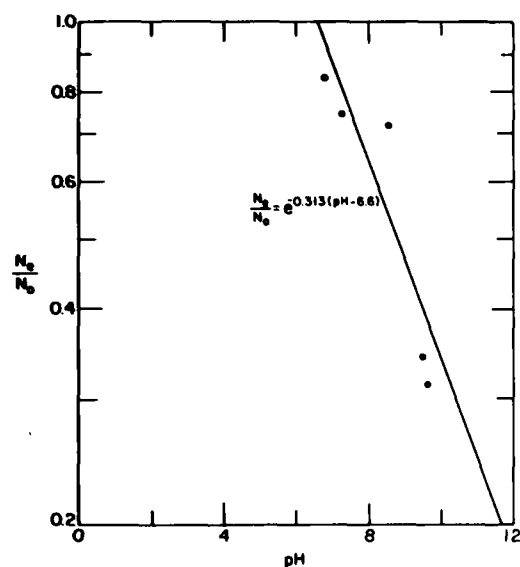


Figure 8. pH vs N_e/N_0 . Data points from Eudora, Peterborough and Corinne, $t \approx 44$ days, average temperature 16.5°C .

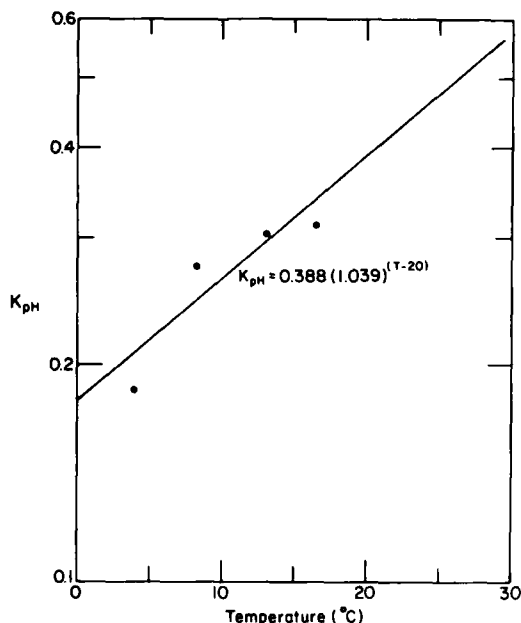


Figure 9. pH rate constant vs temperature.

This equation will predict some nitrogen removal down to a pH of 6.6, which is compatible with the actual ammonia removal data from all of the sites investigated. Nitrogen removal at such a low pH must mean that either some mechanism other than volatilization is responsible or that there are microsites at the water surface with suitable pH even when the bulk liquid gets down to a pH of 6.6.

4. Design equation

It is possible to combine eq 1 and 8, since the temperature dependence is essentially the same for both rate constants:

$$\begin{aligned}
 N_e/N_0 &= e^{-K_T(t+A)} \\
 &= e^{-[0.0064(t) + 0.388 (\text{pH}-6.6)]} \\
 &= e^{-0.0064[t + 60.6(\text{pH}-6.6)]} \\
 &= e^{-K_T [t + 60.6 (\text{pH}-6.6)]}
 \end{aligned}$$

where K_T = temperature-dependent rate constant

$$\begin{aligned}
 K_T &= K_{20^\circ\text{C}} (\theta)^{(T-20)} \\
 K_{20^\circ\text{C}} &= 0.0064 \\
 \theta &= 1.039
 \end{aligned}$$

and all other terms are as defined previously.

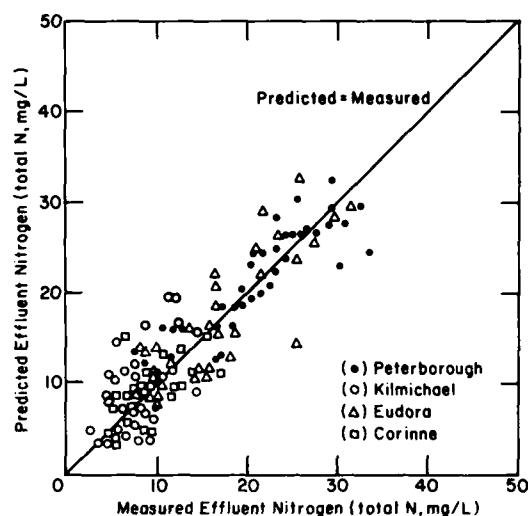


Figure 10. Predicted vs measured effluent nitrogen for all data used in model development. Data range: pH 6.4–9.5, temperature 1.2° to 26.3°C, detention time 27 to 231 days.

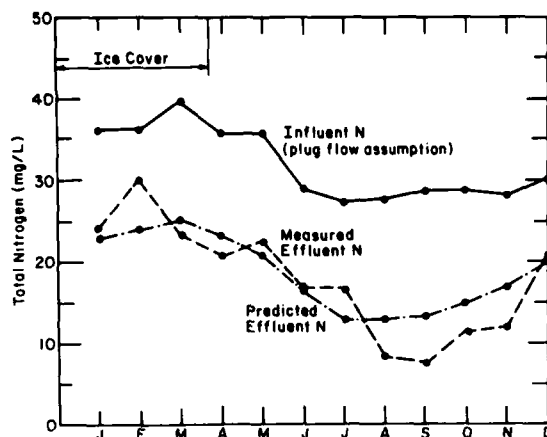


Figure 11. Nitrogen removal in all three cells at Peterborough.

This design equation was applied to all of the 156 data sets and is compared to the values actually measured in the field in Figure 10. A reasonably good fit and a consistent trend are demonstrated. Figures 11–13 show the predicted versus measured effluent values on a monthly basis for the three cases developed for Peterborough, New Hampshire. The influent nitrogen (determined using plug flow assumption) is shown in comparison, as is the period of continuous ice cover.

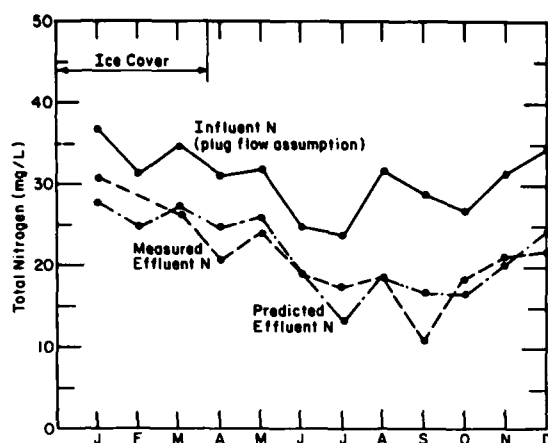


Figure 12. Nitrogen removal in first two cells at Peterborough.

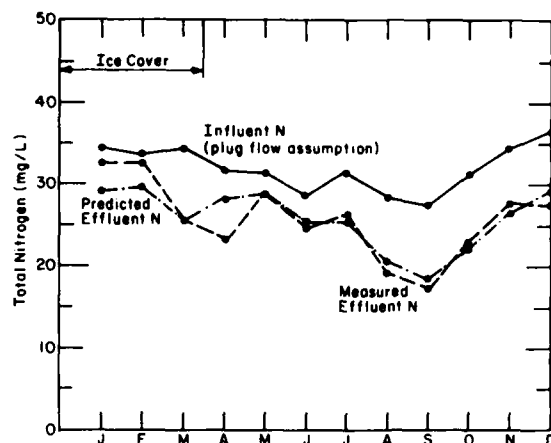


Figure 13. Nitrogen removal in first cell at Peterborough.

5. Model validation

Not all of the 156 data sets available were used in the model development, so to some degree Figure 10 represents at least a partial validation of the design equation. However, it was considered essential to obtain field data from totally different sites and preferably from totally different geographical settings and operating conditions. This is a difficult task since the four systems^{1 10 16 21} used represent the most extensive and most complete data available anywhere in the literature. After considerable searching, a limited amount of data from five treatment ponds were obtained. These were:

Gulfport, Mississippi: single cell, continuous discharge, 5-day detention time polishing pond following aerated ponds, design flow 2900 m³/d.⁵

Drayton Valley, Alberta, Canada: two "long" detention cells, 69 days detention, 5.5 ha, 1.52 m deep, continuous discharge, 1088 m³/d flow, 3550 people.⁹

Stettler, Alberta, Canada: first "long" detention cell, 75 days detention time, 5.34 ha, 1.52 m deep, continuous discharge, 1082 m³/d flow, 3800 people.⁹

Inisfail, Alberta, Canada: first "long" detention cell, 22.6 days detention time, 1.18 ha, 1.52 m deep, continuous discharge, 800 m³/d flow, 2600 people.⁹

Lacombe, Alberta, Canada: single cell, drained in October, refill and storage October to March, discharge continuous and equal to input March to

September. Average detention time during discharge period 165 days, 6.73 ha, 1.52 m deep, 623 m³/d average flow, 3140 people.⁹

Pertinent data from these five sites are tabulated in Appendix B. The effluent values listed are the actual measured results for the time indicated. The other entries were determined using the plug flow assumption previously described. The data from the four Canadian sites represent individual grab samples obtained during the period 1961 to 1964. The values listed for a particular month then represent the average of all samples collected in that same month during the study period (for example, the nine data entries in Table B3 represent 18 separate measurements). There was no data collection in some months and they are not listed.

These five sources yielded a total of 37 data sets that could be used for analysis and model verification. The predicted nitrogen was calculated using eq 9 and the data in Tables B1-B5. These predicted values are compared to the actual effluent nitrogen concentrations in Figure 14. Both the fit and the trend are reasonably close, so it can be concluded that the design equation can be used to give a valid estimate of nitrogen removal in pond systems.

The design equation should therefore be applicable for any typical pond system located in the arctic, subarctic or temperate zone. Figure 15 compares the predicted nitrogen to the actual measured values for the short detention pond in Gulfport.

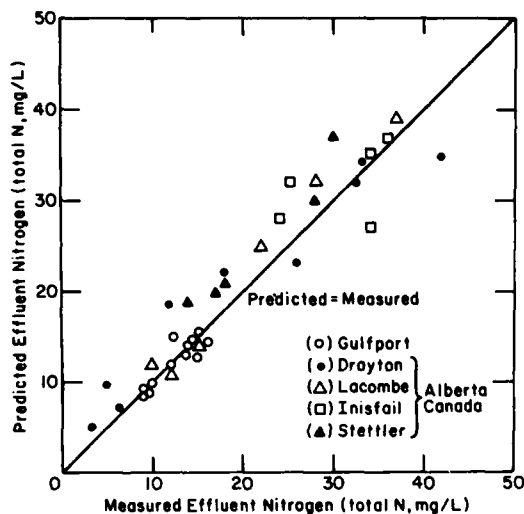


Figure 14. Predicted vs measured effluent nitrogen for all sites used for model verification. Data range: pH 6.9-9.2, temperature 1° to 28°C, detention time 5 to 165 days.

EFFECT OF AN ICE COVER

A continuous ice cover existed at Peterborough for at least three months and at Corinne for a lesser period. However, the design equation will still predict a significant amount of nitrogen removal during these same periods and this is verified by measured values at both sites. Even if all other conditions in the liquid would allow the presence of dissolved ammonia gas it is difficult to see how the gas could escape to the atmosphere from under a continuous ice cover. Some other mechanism may be temporarily responsible for nitrogen removal during periods of ice cover, with volatilization again a factor during ice-free periods.

As might be expected, the ice was quite thick at the Canadian sites examined and approached 1 m or more in depth by the end of the winter (Tables B2-B5). This is at least $\frac{2}{3}$ of the total design depth of the ponds. If about 0.3 m is allowed for sludge accumulation, this means that less than 15% of the design volume was available to actually treat the incoming wastewater. This reduced detention time coupled with the low temperatures is one reason why ice-covered ponds in northern climates perform poorly during the extreme winter months. Effluent quality can be expected to be little better than equivalent to primary under these conditions.

Another water quality effect can be observed due to this ice formation since most of the dis-

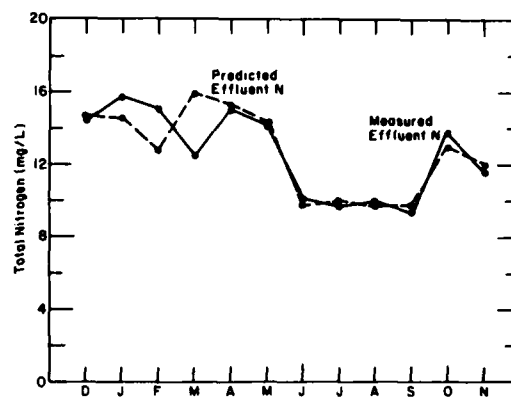


Figure 15. Predicted vs measured effluent nitrogen, polishing pond, Gulfport, Mississippi ($t = 5$ days, pH = 6.9 to 7.6, $T = 10.9^\circ$ to 28°C).

solved constituents in the wastewater will not be incorporated in the ice crystals during the freezing process but will accumulate in the remaining liquid. This has been observed during the freezing of sea ice and has been used as a treatment process for salt reduction in brackish waters.

At the Peterborough site, the alkalinity in the untreated wastewater averaged about 104 mg/L on an annual basis and did not vary much from month to month. During the warm summer months when algal activity was greatest the alkalinity in the pond was about 84 mg/L. The reduction is due to the interaction of algae, CO_2 , and the bicarbonate system previously discussed. During the winter months the alkalinity increased and the investigators attributed the increase to CO_2 production by anaerobic decomposition.¹ However, in February the average alkalinity in the three cells was 164 mg/L, which is about a 60% increase over background, and the low liquid temperatures (3°C) prevailing during the period would not favor significant anaerobic decomposition. An alternate or companion possibility might be concentration effects due to rejection during ice formation.

Similar, but even more dramatic responses can be observed at the Canadian systems used for model validation (Tables B1-B5). At Lacombe, Alberta, for example (Table B5), an ice depth of about 0.5 m is recorded for January and an alka-

linity of 1775 mg/L. This is an increase of about 325% over the average value measured during ice-free periods (575 mg/L). The nitrogen in this pond also increased to 83 mg/L during January as compared to an average of 51 mg/L in the incoming wastewater or an average in the pond of about 16 mg/L during ice-free periods. Since the 0.5 m of ice represents about 30% of the pond volume it seems likely that at least some of these increases are due to the concentrating effects of ice formation.

Fertuck⁹ observed that the ice, after natural freezing of brackish ponds in western Canada, contained about 20% of the salts that were in the original water and he developed a desalination process to take advantage of the phenomenon. Adopting his 20% factor as a "rule of thumb" and further assuming that the ice would have a density of about 0.8 g/cm³, it is possible to calculate the concentration in the remaining liquid for a particular thickness of ice, depth of pond, and concentration of the same material in the original unfrozen water.

It is possible to avoid repetitive calculations and to develop a general expression by normalizing both sides of the equation:

$$\text{Concentration Under Ice/Original Unfrozen Concentration} = f(\text{Ice Thickness/Pond Depth})$$

$$\text{or} \quad C = f(I)$$

where C = concentration increase (%)
 I = ratio of ice thickness to pond depth.

When C equals one, I will equal zero; when I approaches 1.0, C will approach infinity as the concentration increases in the last drop of water just before it freezes. In fact, that last drop would probably not freeze at all due to the increased salt concentration. It is unlikely even in the most extreme climate that the I factor would exceed 0.7 to 0.8 for treatment ponds with a continuous input of relatively warm sewage. An equation can be derived for the boundary conditions described above, with an upper, practical limit for I at 0.8:

$$C = 1 + 1.72 I^{1.56} \quad (10)$$

where C = concentration increase (as a decimal)
 = concentration in remaining liquid (mg/L)/original concentration in unfrozen liquid (mg/L)
 I = ratio of ice thickness (as a decimal)
 = ice thickness/pond depth.

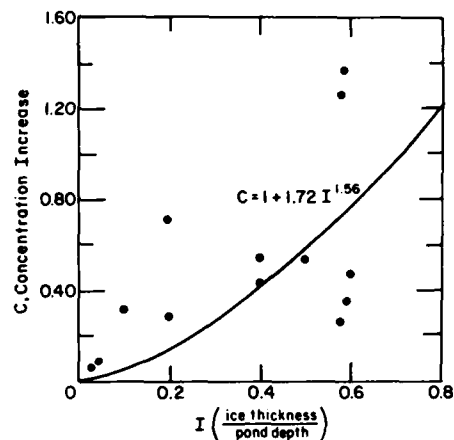


Figure 16. Concentration increase in remaining liquid vs ice cover on a pond. Data points are from Lacombe, Stettler and Drayton Valley.

This expression is plotted in Figure 16 and compared to the observed concentration increases for alkalinity at the Canadian sites included in this analysis. The fit is not statistically significant but eq 10 can provide at least a first approximation of the increase in concentration that might be expected under an ice cover in wastewater ponds. It should be valid for any dissolved constituent that remains dissolved (i.e. no settling, precipitation, etc.).

APPLICATION OF THE DESIGN EQUATION

Input data required for solution of the design equation include the influent nitrogen concentration (N_0), the detention time in the system (t), the temperature of the liquid (T), and the pH of the liquid (pH). Sources and methods for determining these values are described below. Since most regulatory agencies specify 30-day averages as limits it is desirable to repeat the calculation to predict a monthly average effluent nitrogen as well as the annual average. The basic calculation should be concerned with the input to the first cell and the output from the final cell. Three cells are typically used in most current designs and the minimum detention time (t) is that necessary to satisfy the BOD requirements and is determined separately.

1. Determination of input nitrogen

Actual data should be used whenever available, or data from similar facilities. As shown by the ac-

tual data compiled in Appendices A and B the input nitrogen concentration varies significantly over a year's time in typical municipalities. If data are not available, the assumption of a constant average value will be necessary. As indicated previously, the nitrogen in raw domestic sewage ranges to over 50 mg/L. The following values are suggested as preliminary estimates:

	Weak sewage (BOD \approx 120)	Medium sewage (BOD \approx 220)	Strong sewage (BOD \approx 420)
Total N:	20	35	60

If actual monthly averages are available then the input values for calculation should be determined using the plug flow approach previously described.

2. Determination of pond temperature

The temperature in a wastewater pond is a function of pond geometry, detention time, influent temperature and air temperature. The major heat losses are from the water surface by convection and radiation. Eckenfelder⁴ developed the following equation for estimating the bulk liquid temperature of wastewater treatment ponds:

$$T_w = (T_0 + k T_A) / (1 + k) \quad (11)$$

where T_w = mean temperature in pond ($^{\circ}$ F)

T_i = influent temperature ($^{\circ}$ F)

T_A = air temperature ($^{\circ}$ F)

k = proportional factor

$= 1.6 t/d$

t = time (days)

e = pond depth (ft).

It is necessary to use U.S. customary units in solving the equation since the proportionality factor includes a number of coefficients and constants that cannot be easily converted. The equation is not valid after ice formation starts ($T_w = 32^{\circ}$ F). The equation was originally developed for a complete-mix situation but provides a reasonably accurate estimate for plug flow conditions as well. Figure 17 compares the predicted pond temperature using eq 11 to actual values for the Gulfport, Mississippi, pond. The influent temperatures (T_i) can either be obtained from records or estimated by the engineer. The air temperatures (T_A) can be obtained from local weather records. The first iteration should calculate the monthly average water temperature during the plug flow detention period of concern. The average water

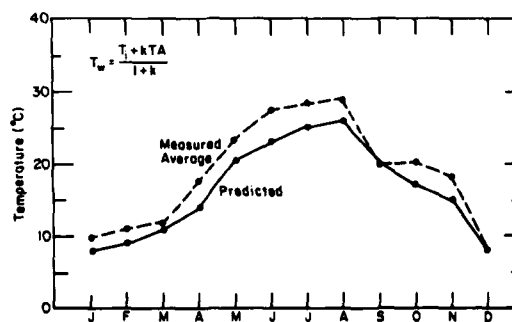


Figure 17. Predicted vs measured pond temperature, Gulfport.

temperature during the plug flow detention period of concern can then be determined from this list.

3. Determination of pH in the pond

The pH of pond contents will vary on a daily basis during warm, sunlit periods, and will seldom be equal to the pH of the entering wastewater due to the interaction between the algae and the CO_2 /alkalinity system in the pond. If possible, actual pH data should be obtained from an operating pond that is subject to conditions similar to those expected for the system under design. The data in Appendices A and B can be used for that purpose. Examination of these data demonstrates the interrelationships of pH and alkalinity. At Peterborough, for example, where the alkalinity was typically less than 100 mg/L, the pH remained about 7. In contrast, at Corinne, where the alkalinity was always above 500 mg/L, the pH was typically 9 or higher.

The relationship between average pH in the overall system and alkalinity in the wastewater was determined for the four systems used in development of the nitrogen removal model:

$$\text{pH} = 7.3 e^{0.0005 \text{ Alk}} \quad (12)$$

where pH = the average pH in the pond
Alk = alkalinity of the entering wastewater (mg/L).

This equation is shown graphically in Figure 18. It will tend to provide a conservative estimate of pH in the pond.

The pond designer will usually initially determine the minimum detention time required for BOD removal. Equation 9 can then be used to estimate the nitrogen removal that will occur under the pH and temperature conditions (determined as

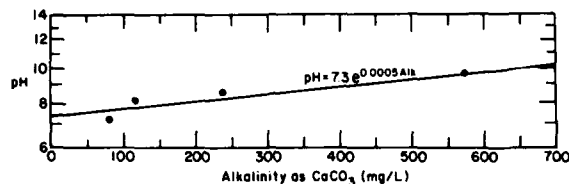


Figure 18. pH in wastewater ponds vs alkalinity. Data points are average observed values from Peterborough, Eudora, Corinne and Kilmichael.

described above) prevailing. If further nitrogen removal is then specifically required, the designer need only consider the nitrogen remaining in the pond effluent. Equation 9 can also be used to determine if additional detention time in the pond can satisfy all of the nitrogen removal requirements. The costs for providing this additional time can then be compared to the costs for alternative nitrogen removal methods. For this purpose, eq 9 takes the following form:

$$t = \ln(N_e/N_0)/(-K_T) - 60.6(\text{pH} - 6.6). \quad (13)$$

All terms were defined previously.

Example: At a pH value of 9 and temperature of 23 °C find the detention time required for 80% nitrogen removal.

$$\begin{aligned} N_e/N_0 &= 0.2 \\ K_T &= K_{20}(\theta)^{(T-20)} = 0.0064(1.039)^{(23-20)} \\ &= 0.0072 \\ t &= \ln(0.2)/(-0.0072) - 60.6(2.4) \\ &= 224 - 145 \\ &= 78 \text{ days for 80\% nitrogen removal.} \end{aligned}$$

The design equation can also be very useful in the design and operation of land treatment systems, since nitrogen is usually the limiting factor determining the land area required for treatment. Under ideal conditions even a few weeks detention in a pond can have a significant effect on nitrogen, and can therefore very significantly influence the size and the costs of the final land treatment component. Recognition of the nitrogen changes described by this equation may also allow selection of more desirable crops or changes in operational procedure. Corn, for example, is a high market value crop, but it has a variable need for nitrogen during the growing season (higher need in early stages). It should be possible to design a system to bypass part of a pond during the early part of the growing season and then use final pond effluent

during the latter stages. This would more closely match the nitrogen needs of the crop and ensure groundwater protection at all times.

A combination treatment/storage pond is often found at land treatment systems in the northern temperate zone. Typically the treatment cell may have 1 to 3 days detention with some aeration (in use, or standby) for odor control. The detention time in the storage cell(s) is determined using procedures in the *Corps of Engineers Land Treatment Design Manual*.²⁶ These storage cells are usually drawn down by late fall and allowed to refill during the cold winter months. A similar concept, termed "controlled discharge," is commonly used in Michigan and adjacent states in the north central United States. In this approach, wastewater is held in the pond during low temperature periods and often also during high algal growth periods, and is discharged once or twice per year when conditions are appropriate in the receiving stream. The Lacombe system used for model verification is a pond of this basic type.⁹

The geometry and hydraulics of these storage cells do not completely satisfy plug flow conditions during the filling and storage periods. The first increment of wastewater during the refilling is likely to spread over the entire pond bottom. If each subsequent increment is assumed to do likewise during the filling period, the contents will approach the complete-mix case since a vertical sample of liquid at any point in the pond will be identical to any other. It is only when the pond starts to discharge that it can be assumed that the incoming wastewater displaces an equivalent volume at the outlet and plug flow is restored.

As a result, the calculation of nitrogen loss from these storage ponds is a two-step procedure. The "average" detention time for use in eq 9 to calculate nitrogen losses during the filling and storage period is half of the actual detention period. The pH and temperature conditions are the averages over the entire period. This will predict the nitro-

gen concentration in the pond just prior to discharge. Once discharge commences, the plug flow assumption prevails. The t is then the total detention time and the nitrogen previously calculated is the N_0 for eq 9 until the pond is drawn down or the contents displaced by fresh wastewater. This approach will allow a conservative estimate of nitrogen removal from any combination of treatment/storage with and without bypasses.

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APPENDIX A: POND DATA USED FOR MODEL DEVELOPMENT

Table A1. Peterborough, three cells, 107 days.

	N_p Final effluent total N (mg/L)	N_o Input, total N (mg/L)	Temp (°C)	pH	Alkalinity (CaCO ₃) (mg/L)
Jan	24.2	33.6	4.3	7.1	71
Feb	30.2	33.9	2.7	7.0	76
Mar	23.1	35.0	2.3	7.0	88
Apr	20.8	33.5	3.0	7.0	100
May	22.5	33.5	8.3	7.0	101
Jun	16.6	29.2	15.0	7.1	91
Jul	16.7	27.4	21.3	7.1	85
Aug	8.5	27.6	23.0	7.1	92
Sep	7.5	28.9	21.3	7.1	90
Oct	11.3	28.9	16.6	7.2	80
Nov	11.9	28.4	11.3	7.2	72
Dec	20.1	30.3	6.7	7.2	71

Table A2. Peterborough, first cell, 44 days.

	N_p Final effluent total N (mg/L)	N_o Input, total N (mg/L)	Temp (°C)	pH
Jan	32.6	34.5	4.2	7.1
Feb	32.6	33.7	4.9	6.8
Mar	25.5	34.4	3.5	6.8
Apr	23.3	31.7	3.3	6.8
May	29.0	31.5	4.9	6.5
Jun	24.9	28.6	11.0	6.6
Jul	25.3	31.6	16.8	6.8
Aug	19.2	28.3	23.6	7.0
Sep	17.2	27.3	22.4	7.3
Oct	23.0	31.2	17.6	7.3
Nov	27.9	34.2	14.5	7.1
Dec	27.8	36.2	8.7	7.2

Table A3. Peterborough, first and second cells, 74 days.

	N_p Final effluent total N (mg/L)	N_o Input, total N (mg/L)	Temp (°C)	pH
Jan	30.9	36.6	4.3	7.1
Feb	33.5	31.3	3.6	6.9
Mar	26.1	34.6	3.1	6.8
Apr	20.7	31.6	3.3	6.8
May	24.3	31.7	5.8	6.5
Jun	19.3	24.9	15.9	6.4
Jul	17.1	23.7	23.6	7.2
Aug	18.5	31.6	23.0	7.0
Sep	10.6	28.9	21.4	7.2
Oct	18.2	26.9	17.2	7.3
Nov	21.2	31.3	14.5	7.2
Dec	21.6	34.2	5.5	7.3

Table A4. Kilimichael, three cells, 214 days.

	N_e Final effluent total N (mg/L)	N_o Input, total N (mg/L)	Temp (°C)	pH	Alkalinity (CaCO ₃) (mg/L)
Jan	9.1	41.2	19.3	8.3	131
Feb	7.8	49.1	17.3	8.2	131
Mar	7.5	47.9	15.0	8.1	127
Apr	5.9	53.5	13.6	8.0	120
May	4.7	40.8	13.6	7.9	112
Jun	4.4	32.9	14.8	8.0	103
Jul	8.1	30.9	17.1	8.1	97
Aug	5.2	24.4	21.9	8.3	98
Sep	5.3	27.3	21.9	8.4	103
Oct	3.9	26.1	23.1	8.4	119
Nov	2.9	34.9	23.4	8.4	119
Dec	9.0	25.1	22.2	8.4	127

Table A5. Kilimichael, first cell, 136 days.

	N_e Final effluent total N (mg/L)	N_o Input, total N (mg/L)	Temp (°C)	pH
Jan	12.0	50.7	15.3	8.0
Feb	11.6	47.1	13.1	8.0
Mar	12.4	37.1	11.1	8.0
Apr	9.0	32.1	11.2	8.0
May	9.2	27.7	13.7	8.1
Jun	7.8	25.9	21.0	8.1
Jul	8.8	26.7	21.0	8.3
Aug	9.0	30.5	24.7	8.4
Sep	9.8	30.0	26.3	8.5
Oct	7.7	45.2	23.5	8.3
Nov	14.8	50.7	19.7	8.2
Dec	14.8	50.7	19.7	8.2

Table A6. Kilimichael, first and second cells, 191 days.

	N_e Final effluent total N (mg/L)	N_o Input, total N (mg/L)	Temp (°C)	pH
Jan	10.6	49.1	18.6	8.4
Feb	8.9	47.9	16.5	8.2
Mar	8.4	53.5	13.4	8.1
Apr	6.7	40.8	12.5	8.2
May	5.7	33.3	13.0	8.2
Jun	4.5	30.9	15.3	8.3
Jul	6.6	24.4	18.4	8.5
Aug	5.8	27.3	21.5	8.6
Sep	8.0	26.1	23.7	8.8
Oct	4.6	34.9	24.6	8.7
Nov	10.4	41.2	24.0	8.7
Dec	9.7	41.2	21.7	8.6

Table A7. Eudora, three cells, 231 days.

	N_e Final effluent total N (mg/L)	N_o Input, total N (mg/L)	Temp (°C)	pH	Alkalinity (CaCO ₃) (mg/L)
Jan	9.1	39.0	17.0	8.4	270
Feb	9.6	48.0	13.9	8.3	280
Mar	9.8	60.5	10.9	8.3	290
Apr	12.7	53.0	8.9	8.3	300
May	8.7	45.0	9.7	8.3	300
Jun	7.7	47.5	10.0	8.3	320
Jul	11.6	49.0	12.6	8.4	300
Aug	10.3	48.5	15.8	8.5	280
Sep	9.0	52.0	18.7	8.6	280
Oct	9.5	52.0	20.7	8.6	270
Nov	10.0	59.0	21.2	8.5	260
Dec	7.7	55.5	19.6	8.5	260

Table A8. Eudora, first cell, 93 days.

	N_e Final effluent total N (mg/L)	N_o Input, total N (mg/L)	Temp (°C)	pH
Jan	23.7	44.0	8.0	7.8
Feb	25.7	51.0	4.7	7.8
Mar	21.6	47.0	4.2	7.9
Apr	31.1	50.0	6.4	8.1
May	29.5	54.0	11.1	8.1
Jun	12.9	50.0	14.7	8.2
Jul	23.5	68.0	21.6	8.2
Aug	25.4	43.0	24.3	8.3
Sep	14.0	35.0	24.6	8.2
Oct	21.5	61.0	22.8	8.2
Nov	27.2	60.0	18.8	8.1
Dec	20.8	46.0	13.2	7.9

Table A9. Eudora, second cell, 44 days.

	N_e Final effluent total N (mg/L)	N_o Input, total N (mg/L)	Temp (°C)	pH
Jan	16.7	25.1	4.5	8.1
Feb	16.5	21.8	3.9	8.3
Mar	15.6	24.3	4.5	8.5
Apr	18.8	24.3	7.0	8.6
May	18.0	24.8	16.3	8.6
Jun	9.8	30.6	22.0	8.5
Jul	15.9	28.0	24.6	8.8
Aug	18.3	16.4	26.7	8.8
Sep	9.1	24.1	24.6	8.5
Oct	14.0	21.6	20.3	8.5
Nov	15.5	17.8	14.6	8.2
Dec	13.3	23.4	8.1	8.0

Table A10. Eudora, first and second cells, 137 days.

	^{N_e} Final effluent total N (mg/L)	^{N_o} Input, total N (mg/L)	Temp (°C)	pH
Jan	16.7	53.0	12.5	8.1
Feb	16.5	45.0	8.2	8.0
Mar	15.6	47.5	5.5	8.0
Apr	18.8	49.0	4.1	8.2
May	18.0	48.5	8.6	8.2
Jun	9.8	52.0	12.7	8.3
Jul	15.9	52.0	17.6	8.4
Aug	18.3	59.0	21.8	8.4
Sep	9.1	55.5	24.0	8.4
Oct	14.0	39.0	23.7	8.5
Nov	15.5	48.0	21.3	8.3
Dec	13.3	60.5	17.0	8.2

Table A11. Corinne, three cells, 42 days.

	^{N_e} Final effluent total N (mg/L)	^{N_o} Input, total N (mg/L)	Temp (°C)	pH	Alkalinity (CaCO ₃) (mg/L)
Jan	7.2	23.4	1.9	9.3	530
Feb	10.6	19.6	1.2	9.0	550
Mar	11.9	14.1	2.8	8.8	530
Apr	8.7	15.3	7.7	9.0	500
May	5.4	16.1	10.0	9.5	510
Jun	5.0	16.1	14.6	9.6	550
Jul	4.1	11.2	20.0	9.4	590
Aug	5.9	8.2	21.1	9.4	600
Sep	7.0	10.3	17.8	9.5	620
Oct	7.2	14.6	12.2	9.5	610
Nov	6.8	14.6	5.7	9.6	550
Dec	4.6	17.6	2.6	9.6	540

Table A12. Corinne, first cell, 27 days.

	^{N_e} Final effluent total N (mg/L)	^{N_o} Input, total N (mg/L)	Temp (°C)	pH
Jan	14.0	23.4	2.1	8.8
Feb	15.4	19.6	1.6	8.4
Mar	12.1	14.1	3.3	8.1
Apr	9.4	15.3	6.9	8.5
May	7.5	16.1	10.1	9.3
Jun	6.9	16.1	14.7	9.5
Jul	6.7	11.2	15.0	9.4
Aug	8.7	8.2	20.7	9.4
Sep	8.2	10.3	17.6	9.4
Oct	8.4	14.6	12.2	9.4
Nov	8.5	14.6	5.7	9.5
Dec	8.0	17.6	2.8	9.5

Table A13. Corinne, first and second cells, 34.5 days.

	^{N_e} Final effluent total N (mg/L)	^{N_o} Input, total N (mg/L)	Temp (°C)	pH
Jan	8.9	23.4	1.9	9.2
Feb	12.3	19.6	1.3	8.8
Mar	12.4	14.1	3.0	8.8
Apr	8.4	15.3	6.2	9.1
May	6.1	16.1	10.0	9.4
Jun	4.9	16.1	14.8	9.6
Jul	5.0	11.2	20.2	9.4
Aug	7.6	8.2	21.0	9.4
Sep	8.2	10.3	17.8	9.5
Oct	7.6	14.6	12.3	9.5
Nov	6.9	14.6	5.8	9.6
Dec	5.8	17.6	2.7	9.6

APPENDIX B: DATA USED FOR MODEL VALIDATION

Table B1. Gulfport, one cell, 5 days.⁵

Date	^{N_g} Final effluent total N (mg/L)	^{N_o} Input, total N (mg/L)	Temp (°C)	pH
Dec 75	14.4	15.4	10.9	7.2
Jan 76	15.8	15.5	12.0	7.2
Feb	15.0	13.5	18.8	7.1
Mar	12.5	16.3	15.3	7.0
Apr	15.0	16.7	21.4	7.2
May	14.0	17.6	22.4	7.6
Jun	10.0	11.5	26.9	7.3
Jul	9.7	10.8	28.0	7.3
Aug	9.9	10.0	28.1	7.2
Sep	9.4	9.7	25.0	7.2
Oct	13.8	13.1	23.6	6.9
Nov	11.6	12.4	18.0	7.1

Table B2. Drayton Valley, final two cells, 69 days (5.5 ha, 1.52 m deep).⁹

Date	^{N_g} Final effluent NH ₄ ⁺ (mg/L)	^{N_o} Input, total NH ₄ ⁺ (mg/L)	Temp (°C)	pH	Alkalinity (mg/L)	Ice depth (m)
Jul 62	5.0	20.6	17.0	8.5	560	0
Aug	3.5	14.3	20.7	9.2	435	0
Nov	6.3	13.5	11.3	9.0	640	0.05
Jan 63	35.0	47.3	1.2	8.3	930	0.76
Mar	33.0	47.8	1.3	8.0	415	0.61
Apr	26.0	31.0	1.8	7.7	500	0.61
May	12.0	28.0	6.9	7.9	465	0
Dec	18.0	32.0	1.3	8.2	790	0.15
Mar 64	42.0	48.0	1.5	7.9	855	0.60

Table B3. Stettler, first long detention cell, 75 days (5.34 ha, 1.52 m deep).⁹

Date*	Final effluent ammonia (mg/L)	Input ammonia (mg/L)	Temp (°C)	pH	Alkalinity (mg/L)	Ice depth (m)
Jul	17	49.5†	20.0	8.7	612	0
Aug	14	49.5†	19.0	8.9	835	0
Nov	37	49.5†	5.0	8.5	870	0
Dec	49	49.5†	1.5	7.9	1115	0.30
Jan	47	49.5†	0.7	7.7	1277	0.91
Feb	55	49.5†	1.0	8.0	1345	0.61
Mar	58	49.6†	1.0	7.6	1030	1.1
May	28	49.5†	12.0	7.7	950	0
Jun	18	49.5†	18.5	8.1	965	0

*Values are averages for month indicated for 1961-64. Data not collected in other months.

†Average of 18 samples, 1961-64.

Table B4. Inisfall, first long detention cell, 22.6 cays (1.18 ha, 1.52 m deep).

Date*	Final effluent ammonia (mg/L)	Input ammonia (mg/L)	Temp (°C)	pH	Alkalinity (mg/L)	Ice depth (m)
Dec	39	47	0.8	7.6	488	0.46
Jan	54	46	1.0	7.5	512	0.66
Feb	46	40	1.1	7.4	525	0.76
Mar	49	40	1.8	7.4	508	0.91
Apr	36	40	2.9	7.4	330	0
May	34	39	8.3	7.5	420	0
Jul	34	37	18.0	8.2	348	0
Sep	24	34	18.0	8.1	380	0
Nov	25	43	12.0	7.9	415	0.03
Average 42.2						

*Values are averages for month indicated for 1961-64. Data not collected in other months.

Table B5. Lacombe, one cell (6.73 ha, 1.52 m deep). Drained in late October, refill and storage October-March; discharge equal to input March to September.

Date*	Final effluent ammonia (mg/L)	Input ammonia (mg/L)	Temp (°C)	pH	Alkalinity (mg/L)	Ice depth (m)
Jan	83	51.25†	0.73	8.2	1775	0.46
Feb	57	51.25†	1.0	7.8	882	0.88
Mar	39	51.25†	1.0	7.9	963	0.91
Apr	28	51.25†	5.0	7.4	555	0
May	12	51.25†	24.0	9.4	600	0
Jul	15	51.25†	18.0	8.3	535	0
Aug	10	51.25†	17.0	9.3	490	0
Nov	22	51.25†	4.0	8.1	760	0.06
Dec	47	51.25†	1.0	7.9	1200	0.30

*Values are averages for month indicated for 1961-64. Data not collected in other months.

†Average of four samples of raw sewage taken in 1963-64.

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Reed, Sherwood C.

Nitrogen removal in wastewater ponds / by Sherwood C. Reed. Hanover, N.H.: Cold Regions Research and Engineering Laboratory; Springfield, Va.: available from National Technical Information Service, 1984.

v, 33 p., illus.; 28 cm. (CRREL Report 84-13.)

Prepared for Office of the Chief of Engineers by Corps of Engineers, U.S. Army Cold Regions Research and Engineering Laboratory under Da Project 4A762730AT42.

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